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Isolating the effect of soil properties on agricultural soil greenhouse gas emissions under controlled conditions

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Abstract

Agricultural soils are important sources of greenhouse gases (GHGs). Soil properties and environmental factors have complex interactions which influence the dynamics of these GHG fluxes. Four arable and five grassland soils which represent the range of soil textures and climatic conditions of the main agricultural areas in the UK were incubated at two different moisture contents (50 or 80 % water holding capacity) and with or without inorganic fertiliser application (70 kg N ha^{-1} ammonium nitrate) over 22 days. Emissions of N_2O, CO_2 and CH_4 were measured twice per week by headspace gas sampling and cumulative fluxes were calculated. Multiple regression modelling was carried out to determine which factors (soil mineral N, organic carbon and total nitrogen contents, C:N ratios, clay contents and pH) that best explained the variation in GHG fluxes. Clay, mineral N and soil C contents were found to be the most important explanatory variables controlling GHG fluxes in this study. However, none of the measured variables explained a significant amount of variation in CO_2 fluxes from the arable soils. The results were generally consistent with previously published work. However, N_2O emissions from the two Scottish soils were substantially more sensitive to inorganic N fertilisation at 80% water holding capacity than the other soils, with the N_2O emissions being up to 107 times higher than the other studied soils.

Keywords: GHG emissions, inorganic fertiliser, agricultural soils

Running head: Agricultural soil greenhouse gas emissions

1 Introduction

Agricultural soils are important sources of atmospheric greenhouse gases (GHGs). Various soil properties, environmental factors and management practices have complex interactions which influence the dynamics of these GHG fluxes. Soil texture is a particularly important factor as it dictates soil water dynamics, pore space and gas diffusivity (Skiba & Ball 2002). The availability of nitrogen (N) for microbial processes is also an influential factor (Cardenas *et al.* 2019).

Greenhouse gas fluxes from agricultural soils are influenced by environmental factors such as temperature, precipitation, and soil physical and chemical properties such as texture, pH, oxygen concentration and nutrient availability. Texture affects pore space distribution and gas diffusivity (Smith *et al*, 2003) whilst soil pH manipulates the microbial community structure, and therefore the decomposition or accumulation of soil organic carbon (SOC) (Malik *et al*, 2018). The soil texture, particularly the clay content, determines the level of physico-chemical stabilisation of SOC through association with soil minerals (Schrumpf *et al*, 2013).Soil wetness strongly influences soil GHG emissions. Production of N_2O by nitrification increases linearly as increasing soil water content approaches 60% of water filled pore space (WFPS). At higher water contents denitrification becomes more prevalent leading to maximum emissions at around 80% WFPS (Shepherd, 2009). Soil CO_2 emissions decrease substantially after heavy rainfall because poor gas diffusivity and low air-filled porosity restrict respiration and increase anaerobic conditions (Ball, 2013). Anaerobic soil conditions can promote the production of CH_4 via methanogenesis whilst methanotrophy (CH_4 oxidation to CO_2) is more prominent in aerated soils allowing diffusion of CH_4 into the soil from the atmosphere (Cloy and Smith, 2015). In fine textured soils, pore spaces are smaller and so a lower volume of water is required to reach the same WFPS as in a coarser textured soil. Limited diffusion in fine textured soils therefore tends to support the development of anaerobic microsites and so tend to emit more N_2O than coarser textured soils (Stehfest and Bouwmann, 2006).

Application of inorganic N such as ammonium nitrate (AN) fertiliser to soil temporarily creates an excess of available-N required for microbial nitrification and denitrification (NH_4^+ and NO_3^-, respectively) which reduces microbial competition for these resources (Norton and Firestone, 1996).

The application of inorganic N may also decrease or reverse the soil's CH_4 sink and source capacity (Inselsbacher *et al*, 2011).

Across the UK, GHG emissions from agricultural soils vary widely as a consequence of climate, management and soil type. The objective of this study was to isolate the effect of soil chemical and physical properties on GHG emissions by measuring GHG fluxes from soils in a controlled environment. Soils were subjected to two different moisture contents with or without AN application.

2 Materials and Methods

2.1 Soils

Soils were collected from nine UK Agricultural Greenhouse Gas Research Platform sites: four arable (Boxworth, Gilchriston, Rosemaund and Woburn) and five grassland (Crichton, Drayton, Hillsborough, North Wyke and Pwllpeiran). The soils did not receive any N inputs from the end of the 2010 growing season to collection (February to early March 2011, McGeough *et al*, 2016). Soils were sieved to < 4 mm to remove large stones and roots, air dried and stored in sealed plastic bags. These sites represent the different soil types and climates of the main agricultural areas across the UK (Table 1).

2.2 Treatments

A fully-factorial experiment was designed with two water holding capacities (WHCs) (50% and 80%), and two AN application levels (0 or 70 kg N ha^{-1}). The method of Howard and Howard (1993) and the following equation were used to determine WHC:

$$100\%\ \text{WHC} = \frac{\text{mass saturated soil - mass oven dry soil}}{\text{mass oven dry soil}}$$

Treatments were applied in triplicate to 80 g of soil at a bulk density of ~1 g cm^{-3} (average value found from field measurements of these soils: range 0.6–1.6 g cm^{-3}), in 500 ml Kilner jars. Ammonium nitrate was dissolved in the deionised water used to adjust the WHC. Soils were incubated at 10 °C (average annual temperature for all sites, Table 1) for 22 days, following a three day pre-incubation period.

2.3 Headspace gas sampling

Headspace gas sampling was undertaken twice per week. Gas samples were taken from Kilner jars at the beginning (t0) and end (t1) of a one hour closure period. Before each sampling period, jars were opened for three minutes to allow gas concentrations in the jar to equilibrate with the laboratory air before sealing the lids with both sampling ports open. The jars were flushed three times through one sampling port using a 60 ml syringe before drawing a 30 ml t0 gas sample and injecting it into a 25 ml pre-evacuated vial, after which both ports were closed. After the t0 headspace gas sampling, jars were returned to the incubator and the 30 ml t1 samples were drawn one hour later with one port remaining closed. Between sampling periods jar lids were closed with both ports open to allow free gas exchange whilst limiting moisture loss. Moisture loss was never more than 1%, and so was not deemed to be significant (calculated from mass change of kilner jars from beginning to end of incubation).

2.4 GHG calculations

Gas samples were analysed for N_2O, CO_2 and CH_4 using an Agilent 7890A Gas Chromatograph (GC) fitted with electron capture, flame ionisation and thermal conductivity detectors (Agilent Technologies, Berkshire, UK) and a CTC Analytics COMBI PAL autosampler (CTC Analytics, Hampshire, UK). The GC gas peak area responses were calibrated (calibration curves were linear for CO_2 and CH_4, quadratic for N_2O) using four certified standard gas mixtures (BOC Industrial Gases, UK). Headspace GHG concentrations were used to calculate GHG fluxes per day using linear regression and the ideal gas law (Saggar *et al*, 2008):

$$\mathrm{F} = \rho \frac{V}{A} \; \frac{\Delta C}{\Delta t} \frac{273.15}{T + 273.15}$$

This calculation assumes a linear increase in gas concentration in a known volume over a known period of time, where F = flux, ρ = gas density, V = jar volume, A = jar basal area, Δc = difference between gas concentrations at t1 and t0, Δt = jar closure time (hours), T = incubation temperature (°C).

Cumulative fluxes were calculated using the trapezoidal rule (area under the curve) to interpolate fluxes between sampling days (Hinton *et al*, 2015; Bell *et al* 2015a,b; Bell *et al*, 2016) as follows:

Cumulative flux = (day *x* cumulative flux + day *y* flux) + (mean (day *x* flux + day *y* flux))

* (day *y* - day *x* - 1)

Emission factors (EFs) define the percentage of applied N fertiliser which is emitted as N_2O. Emission factors were calculated for N_2O emissions from AN fertilised soils incubated over the 22-day incubation period using the following equation:

$$\mathrm{EF} = \left(\frac{\mathrm{FN_2O\ flux\ \ (kg\ N_2O - N) - CN_2O\ flux\ (kg\ N_2O - N)}}{N\ applied\ (kg\ N)}\right) * 100$$

where FN_2O = cumulative N_2O flux from fertilised soil and CN_2O = cumulative N_2O flux from unfertilised control soil.

Global warming potentials (GWPs) were calculated as CO_2 equivalents (CO_2-eq) using IPCC (2014) values over a 100-year timescale of 1, 28 and 265 for CO_2, CH_4 and N_2O respectively.

2.5 Chemical analyses

Pre- and post-incubation soil mineral N concentrations were determined. Soil subsamples were extracted with 2 M KCl (1:2 soil to KCl) within 24 hours of the final headspace gas sampling. Extracts were analysed for NH_4^+-N and NO_3^--N using a Skalar San^{++} continuous flow colorimetric autoanalyser (Skalar, York, UK). Colorimetric determination was carried out at wavelengths of 650 nm and 540 nm for NH_4^+-N and NO_3^--N, respectively, following the methods of Singh *et al* (2011).

Pre-incubation soil subsamples were extracted with deionised water (1:2) and soil solution pH was measured using a calibrated pH electrode (Thermo-Orion, Beverly, MA, USA).

Air dried, ball milled soil samples were combusted and analysed for organic C (OC) and total nitrogen (TN) using a Flash 2000 elemental analyser (Thermo Fisher Scientific, Bremen, Germany).

2.6 Statistical Analyses

Treatment effects (WHC and AN fertilisation) on cumulative GHG fluxes and pre- and post-incubation mineral N contents were determined by two-way ANOVA. Significant differences between EFs of different soils were determined by one-way ANOVA. Multiple linear regressions were used to evaluate the influence of % clay (as a proxy for soil texture), % OC, % TN, C:N, pH and NH_4^+-N and NO_3^--N content on arable and grassland soils. Reduced models were determined by backwards selection of the most significant variables. All statistics were carried out using Genstat (15th edition). To more fully satisfy the assumption of normal distribution of the residuals data was log transformed where appropriate and outlying cumulative GHG flux points were removed from the analyses after scrutiny of the residuals.

3 Results

3.1 Soil properties

Ranges of soil properties across grassland soils were as follows: average TN content 0.27–0.77%; average OC content 2.22–12.35%; average clay content 15.0–56.5%; C:N ratio 8.17–15.99 and pH 4.47–6.67. Ranges of soil properties across arable soils were as follows: average TN content 0.09–0.19%; average OC content 0.94–1.90%; average clay content 11–45%; C:N ratio 8.21–13.57 and pH 5.13–7.91. Specific values for each soil are given in Table 1. Average clay contents were taken from McGeough *et al* (2016). These differences in soil properties are due to natural variation in geology, topography and climate.

3.2 GHG fluxes

Global warming potentials (CO_2-eq) for grassland and arable soils (Table 2) show that the GHG budgets were generally dominated by N_2O and CO_2 fluxes.

3.2.1 N_2O emissions

The Scottish arable and grassland 80% WHC+N treatment soils had substantially higher N_2O emissions than the other soils ($p < 0.05$, Figure 1a,b). The Scottish grassland soil 80% WHC treatment was also significantly higher than other treatments in all soils except one (Pwllpeiran) ($p < 0.05$, Figure 1b).

During the 22-day incubation period N_2O EFs calculated for the arable and grassland soils (Figures 2a, b) were consistently below the IPCC default value of 1%. This was expected since EFs are usually calculated from one year field measurements however, the EFs calculated here are useful for site comparisons. At 50% WHC the EFs were negligible. The Scottish soils had significantly higher EFs ($p < 0.01$) than the other soils at 80% WHC.

The negative EFs observed for Boxworth and Crichton 50% WHC do not indicate uptake of N_2O from the atmosphere, but rather that the emissions from the unfertilised control treatments were greater than from the fertilised treatments. There were no significant differences between emissions from fertilised and unfertilised soils in these cases where negative EFs were observed.

3.2.2 CO_2 fluxes

There was high variability in cumulative CO_2 fluxes between replicates in the grassland and arable soils. For instance, CO_2 fluxes from Crichton soils at 50% WHC were 2760 ± 1450 mg CO_2-C m^{-2}, Hillsborough soils at 50% WHC+N had fluxes of 945 ± 532 mg CO_2-C m^{-2}, at 80% WHC Drayton soil had fluxes of 8890 ± 2350 CO_2-C m^{-2} and the 80% WHC+N North Wyke soils had fluxes of 382 ± 3400 CO_2-C m^{-2} (Figure 3a). Measured apparent negative or zero CO_2 fluxes are considered to be due to analytical constraints near the detection limit of the GC.

3.2.3 CH_4 fluxes

The CH_4 fluxes calculated for each sampling day provided evidence that both methanogenesis and methanotrophy were occurring simultaneously in all soils with some alternating strongly between being a source and a sink (Figure 4a,b). Calculated cumulative fluxes can be assumed to reflect the dominant process in each soil and treatment combination. Cumulative CH_4 fluxes were highly variable within arable soils (Figure 4a).

3.3 Mineral N concentrations

For all soils there were large differences between initial untreated pre-incubation soil NO_3^--N contents, but not corresponding NH_4^+-N contents (Figure 5). Unfertilised and AN fertilised post-incubation Hillsborough and North Wyke grassland soils exhibited greatest loss or microbial transformation of native soil NO_3^--N and added fertiliser NO_3^--N. Results for the unfertilised post-incubation Crichton grassland soils suggest net production of NH_4^+-N via OM mineralisation.

Fertilisation with AN had a significant effect on NO_3^--N and NH_4^+-N contents for both grassland and arable soils ($p < 0.001$) with increases of 5 (50% WHC) and 25 (80% WHC) times relative to unfertilised soils being observed.

3.4 Bivariate Correlations

Pre-incubation NO_3^--N contents were positively correlated with OC, TN and pre-incubation NH_4^+-N contents and negatively correlated with soil ph. Soil TN and pH were positively correlated with clay content and C:N ratio was negatively correlated with clay content (Table 3). Soil C:N ratio was correlated positively with pH and C:N, OC and TN contents were all positively correlated to each other.

3.5 General Linear Modelling

Results from both full and reduced models are shown for arable (Table 4) and grassland (Table 5) soils. For all models, except for the arable cumulative CO_2 flux, the full model explained 1–2% more of the variation than the reduced model. For arable cumulative N_2O fluxes, the reduced model explained 60% of the variation with significant positive correlations with OC ($p < 0.01$) and NO_3^--N ($p < 0.05$) contents, and negative correlation with clay contents ($p < 0.01$). For grassland cumulative N_2O fluxes the reduced model explained 69% of the variation with significant negative correlations with NO_3^--N ($p < 0.01$) and clay ($p < 0.01$) contents and positive correlation with NH_4^+-N ($p < 0.01$) contents.

The negative correlation between N_2O flux and clay content seems to be atypical and is caused by the high emissions and low clay contents of the two Scottish soils (Gilchriston and Crichton). When these two soils were removed from the analysis the relationship became positive.

It was not possible to adequately describe the variation of cumulative CO_2 fluxes from the arable soils with the measured variables. CO_2 fluxes from arable soils were much more variable from day to day than from grassland soils. The full model described only 2% of the variation. The reduced models explained 52% of the variation for cumulative CO_2 fluxes from the grassland soils. Cumulative CO_2 fluxes had significant ($p < 0.01$) positive correlations with NH_4^+-N and clay contents and significant negative correlation with NO_3-N contents.

The reduced model for arable cumulative CH_4 fluxes explained 37% of the variation with significantly positive correlations with TN contents ($p < 0.01$) and % WHC ($p < 0.05$), and negative correlation with clay contents ($p < 0.001$). The reduced model for grassland cumulative CH_4 fluxes explained 18% of the variation with a significant positive correlation with OC contents ($p < 0.01$).

4 Discussion

4.1 N_2O fluxes and emission factors

In this study, all soils were processed and incubated in the same way but the two Scottish soils displayed substantially higher N_2O fluxes than the other soils, particularly at the higher moisture content. However, laboratory and field studies investigating the soils from these sites have reported varying results. McGeough *et al* (2016) found much higher N_2O emissions from the Scottish grassland soil studied here (60% WFPS, incubated at 15 °C for 60 days, 100 μg N g^{-1} dry soil), although not from the Scottish arable soil. In field trials, Bell *et al* (2015b) found that the annual EF for the Scottish arable soil was ~3 to 5 times higher than grassland sites elsewhere in the UK. However, Cardenas *et al* (2019) did not observe higher annual EFs from the Scottish grassland site in field measurements when compared to other UK grassland sites across a range of N application rates (80 - 400 kg N ha^{-1}).

In these controlled incubations, clay, mineral N and OC contents were found to be the most important factors in determining N_2O fluxes from arable and grassland soils. Clay content (and therefore texture) has frequently been identified as an important factor controlling soil N_2O emissions (Skiba and Ball, 2002; Dobbie and Smith, 2003; Stehfest and Bouwmann, 2006). Positive correlations between N_2O flux and OC content have also previously been observed (Stehfest and Bouwmann, 2006).

In this study, EFs were consistently below the value assumed in the IPCC Tier 1 methodology (1%) (IPCC, 2006a). It should be noted that the 22 day incubation period used in this study was much shorter than the 12 months normally used to assess EFs, however, it does provide a valuable ranking of the proportion of emissions that can be attributed to the added N source.

Field experiments at the Scottish sites have also reported low EFs in the short term after AN application. Hinton *et al* (2015) found EFs from 0.44–0.56% for the five weeks following AN application, but annual EFs of 1.36, 0.96 and 1.08% for AN application rates of 120, 160 and 200 kg N ha^{-1} at the Scottish arable site. Smith *et al* (2012) found an EF of 0.61% (over the growing season) at the Scottish grassland site in 2004, however, the EF in 2003 was higher (1.13%). EFs are highly variable, values between 0.2 and 7% have been reported for agricultural fields in Scotland (Clayton *et al,* 1997; Smith *et al*, 1998; Dobbie *et al*, 1999) and the range of uncertainty associated with the IPCC default value is 0.3–3.0% (IPCC, 2006b).

It has been suggested that the higher EFs from Scottish sites is due to the incidence and intensity of rainfall (and therefore WFPS) at the time of fertiliser application (Dobbie *et al,* 1999). However, this does not explain the higher EFs from Scottish soils under controlled conditions. Soil OC stocks are higher in Scottish agricultural soils (compared with elsewhere in the UK) (Bradley *et al*, 2005) and so the distribution and availability of soil OC pools may differ. Further investigation of OC pools and their availability, aggregate-stabilising minerals and microbial communities within the UK soils studied here may explain these unexpected findings.

4.2 CO_2 fluxes from UK arable and grassland agricultural soils

A positive correlation between grassland CO_2 fluxes and clay content was found in this study, which is counter to the theory that higher clay contents provide a greater opportunity for chemical protection of OM by adsorption. Dilustro *et al* (2005) also observed greater CO_2 fluxes from clay textured (> 19% clay) than sandy textured (< 12% clay) forest soils. However, they attribute this to a more dense vegetation (and so greater root respiration) on the clay soils and the sandy soils being excessively drained for part of the study period.

In this study, grassland soil CO_2 flux showed a positive correlation with NH_4^+-N content and a negative correlation with NO_3^--N content. However, fertilisation with AN was observed to decrease or have no effect on CO_2 emissions. Zaman *et al* (2002) speculate that fertilisation without the addition of C cannot drive increased microbial growth or respiration. There are several conflicting results in the literature which show increases (Baggs *et al*, 2003), decreases (al-Kaisi *et al*, 2008) and no effect (Baggs *et al*, 2003; Garcia-Ruiz and Baggs, 2007 and al-Kaisi *et al*, 2008) of AN fertilisation on CO_2 flux.

4.3 CH_4 fluxes from UK arable and grassland agricultural soils

Individual soils showed highly variable CH_4 fluxes throughout the incubation period, the oscillation of fluxes from net source to net sink indicates that methanogenesis and methanotrophy were occurring simultaneously (Ekberg and Christensen, 2006). As a result the net emissions of CH_4 when expressed as CO_{2e} was small relative to the other greenhouse gases. Soil moisture content, native N content and clay content explained significant variation in emissions from arable soils and OC content from grassland soils.

Soils with coarse textures have higher oxidation rates of CH_4 than more fine textured soils, attributable to low porosity and high water retention in fine textured soils causing low gas diffusivity into the soil (Dörr *et al*, 1993; Dutaur and Verchot, 2007; Tate *et al*, 2007).

In this study increasing the moisture content from 50 to 80% WHC actually reduced emissions (increased sinks) in all cases except one. It is possible that the methanotrophic microbial population was under water stress at the lower WHC level. von Fischer *et al* (2009) found that methanotrophic activity dropped off sharply below 40% WFPS in a sandy loam grassland soil. The presence of NO_3^--N would also act as an inhibitor to methanotrophs as this would be used in preference to organic carbon as a terminal electron acceptor.

4.4 Overall GHG budget

When expressing the GHG fluxes measured from incubated soils in this study in terms of their GWPs, it is clear that the GHG budget is driven by N_2O and CO_2 fluxes. However, under field conditions much of the CO_2 released from soil by respiratoin is returned by photosynthesis and so may not be a net source of atmospheric CO_2. This highlights the importance of accurately assessing the effects of agricultural soil management on N_2O emissions (Gao *et al.* 2018).

Conclusions

Generally, the results were in support of those found in the literature for a wide range of soils, conditions and locations with soil texture, soil mineral N and OC contents found to be the most important measured variables controlling GHG fluxes. However, the N_2O emissions from Scottish soils were more sensitive to ammonium nitrate fertilisation, particularly at 80% WHC, than the other UK agricultural soils studied here. The reason for the high EFs from Scottish soils remains unclear, however, it is possible that it could be linked to differences in the structure of the microbial population or composition of the soil organic matter pools. Resolving this issue would be valuable in making more precise predictions of N_2O emissions in response to soil management.

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Table 1

Land Use	Site	Average annual temp. (°C)	Soil texture	Annual rainfall (mm)	100% WHC (g water g^{-1} soil)	TN (g N kg^{-1} soil)	OC (g C kg^{-1} soil)	C:N ratio	Clay (% by weight)	pH
Grassland	Crichton	10.2	Sandy loam	> 950	0.55 (0.03)	3.06 (0.163)	32.6 (1.52)	10.68 (0.046)	12.5	5.3 (0.03)
	Drayton	10.3	Clay	0 - 750	0.51 (0.04)	4.62 (1.383)	50.8 (15.60)	11.01 (0.477)	50.1	6.6 (0.42)
	Hillsborough	9.8	Clay loam	751 - 950	0.34 (0.04)	7.72 (0.107)	12.4 (0.77)	15.99 (0.077)	28.1	5.9 (0.03)
	North Wyke	9.6	Silty clay	> 950	0.64 (0.03)	2.72 (0.060)	22.2 (0.83)	8.17 (0.089)	32.5	4.5 (0.02)
	Pwllpeiran	9.3	Clay loam	> 950	0.70 (0.16)	4.41 (0.638)	41.8 (6.06)	9.49 (0.035)	23.8	5.1 (0.10)
Arable	Gilchriston	8.7	Sandy Clay loam	< 750	0.45 (0.02)	0.93 (0.023)	12.7 (0.53)	13.57 (0.213)	12.7	5.9 (0.03)
	Woburn	10.9	Loamy sand	< 750	0.45 (0.04)	0.86 (0.037)	9.4 (0.40)	10.93 (0.027)	10	7.0 (0.08)
	Rosemaund	10.4	Clay loam	751 - 950	0.30 (0.05)	1.28 (0.062)	10.5 (0.18)	8.21 (0.163)	20.9	5.1 (0.02)
	Boxworth	9.7	Clay	550	0.51 (0.00)	1.85 (0.043)	19.0 (0.35)	10.27 (0.036)	44.8	7.9 (0.05)

Table 2

Treatment	Land Use	Site	N_2O	CO_2	CH_4
50% WHC	Arable	Boxworth	651	1460	10.97
		Gilchriston	477	985	4.07
		Rosemaund	-41.0	301	12.25
		Woburn	21.1	-90.6	5.95
	Grassland	Crichton	884	2760	-0.82
		Drayton	298	2040	-33.28
		Hillsborough	-61.4	140	33.34
		North Wyke	128	1060	-0.41
		Pwllpeiran	99.9	-168	-6.69
50% WHC+N	Arable	Boxworth	394	2090	20.32
		Gilchriston	570	1740	4.58
		Rosemaund	-49.1	2770	32.18
		Woburn	-2.8	1840	0.35
	Grassland	Crichton	495	2750	-21.1
		Drayton	331	3720	10.38
		Hillsborough	-8.4	945	21.87
		North Wyke	169	1990	-4.07
		Pwllpeiran	-13.9	-263	17.29
80% WHC	Arable	Boxworth	463	1130	6.56
		Gilchriston	1450	1440	-7.1
		Rosemaund	-81.8	1980	5.28
		Woburn	75.3	363	-26.06
	Grassland	Crichton	1300	4780	-19.66
		Drayton	411	8890	-8.64
		Hillsborough	109	2550	26.71
		North Wyke	297	3390	-1.35
		Pwllpeiran	41.6	437	-21.29
80% WHC+N	Arable	Boxworth	1436	1560	21.48
		Gilchriston	12300	1220	-6.43
		Rosemaund	-56.9	891	38.04
		Woburn	114	-69.9	-20.11
	Grassland	Crichton	182000	1780	-20.12
		Drayton	1840	6390	1.54
		Hillsborough	127	3810	12.57
		North Wyke	1230	382	-13.4
		Pwllpeiran	74.5	938	-2.71

Table 3

Variate	Clay (% by weight)	OC (% by weight)	TN (% by weight)	C:N ratio	pH	NH_4^+-N (mg kg^{-1})	NO_3^--N (mg kg^{-1})
clay	1.00						
OC	0.19	1.00					
TN	**0.33 ***	**0.96 ***	1.00				
C:N	**-0.21 ***	**0.69 ***	**0.51 ***	1.00			
pH	**0.26 ***	-0.06	-0.16	**0.27 ***	1.00		
NH_4^+-N	0.00	0.02	0.01	0.01	0.02	1.00	
NO_3^--N	0.09	**0.27 ***	**0.31 ***	0.00	**-0.23 ***	**0.89 ***	1.00

Table 4

	Arable CO_2				Arable N_2O				Arable CH_4			
	Full Model		Reduced Model		Full Model		Reduced Model		Full Model		Reduced Model	
Variable	CV	SE	CV	SE	CV	SE	CV	SE	CV	SE	CV	SE
NO_3N content	-1732	4.33			0.00048	0.000671	**0.0003** ****	0.0000				
NH_4^+-N content	-2.43	4.36			-0.000178	0.000684			-0.0003	0.0002		
% WHC	19	15.4			0.0061	0.0032			**0.0146** ****	0.0065	**0.0146** ****	0.0066
OC content	3024	2535	1082	604	**2.798** *	0.486	**2.723** *	0.328				
TN content	-19479	23840							**94.4** **	28.2	**72.9** *	18.5
C:N ratio									0.0851	0.0827		
Clay content					**-0.0658** *	0.013	**-0.0638** *	0.0089	**-0.243** **	0.078	**-0.1855** *	0.0535
pH												
Constant	-1732	1458	-392	818	**-2.361** *	0.454	-1.774	0.259	**-7.73** **	2.56	**-5.59** *	1.2
F	p = 0.328		p = 0.08		*		*		*		*	
R^2	0.02		0.05		0.61		0.60		0.39		0.37	

Table 5

	Grassland CO_2				Grassland N_2O				Grassland CH_4			
	Full Model		Reduced Model		Full Model		Reduced Model		Full Model		Reduced Model	
Variable	CV	SE	CV	SE	CV	SE	CV	SE	CV	SE	CV	SE
NO_3N content	7.32	8.73	**-6.26 ***	1.24	**-0.0022 ****	0.0007	**-0.0018 ***	0.0002	0.0030	0.0035		
NH_4^+-N content	-4.46	8.86	**9.32 ***	1.41	**0.0028 ***	0.0007	**0.0023 ***	0.0002	-0.0033	0.0035		
% WHC	8.8	17.1			0.003	0.0024			0.0120	0.0067		
OC content					-0.094	0.114			0.008	0.437	**0.1036 ***	0.0279
TN content	-39464	24582			2.26	3			-8.08	8.96		
C:N ratio	2108	1239							0.549	0.608		
Clay content	**113.9 ***	33.6	**88.7 ***	19.5	-0.00537	0.00427	**-0.008 ****	0.0028				
pH	2127	1458			-0.206	0.206			0.253	0.564		
Constant	-22114	12672	526	818	1.466	0.608	**0.7506***	0.0806	-5.36	5.99	**-0.509 ****	0.184
F	*****		*****		*****		*****		*******		*****	
R^2	0.54		0.52		0.70		0.69		0.20		0.18	

Table 1: Land use, average annual temperature (°C), soil texture, annual average precipitation for the nine soils**, p**re-incubation average 100% water holding capacity (WHC), total nitrogen (N) and, organic carbon (OC) contents, C:N ratios, clay contents and pH for the grassland and arable agricultural soils.

Table 2: Global warming potentials (CO_2-eq) of grassland and arable soils incubated over a 22 day period at 50 or 80% water holding capacity (WHC) and with or without ammonium nitrate (N) fertiliser.

Table 3: Bivariate correlations between measured soil properties (pre-treatment) for all nine soils. Clay, organic carbon (OC), total nitrogen (TN) NH_4^+-N and NO_3^--N contents, pH and C:N ratio. * Significant correlations ($p < 0.05$).

Table 4: Full and reduced multiple linear regression models for arable soils. Variables assessed were NO_3^--N and NH_4^+-N contents (mg kg^{-1}), % water holding capacity (WHC), organic carbon (OC) and total nitrogen contents (TN) (% by weight), C:N ratios, clay content (% by weight) and pH. CV is the coefficient of variance, SE is the standard error. Significance levels are denoted as follows: * $p < 0.001$, ** $p < 0.005$, *** $p < 0.01$, **** $p < 0.05$.

Table 5: Full and reduced multiple linear regression models for grassland soils. Variables assessed were NO_3^--N and NH_4^+-N contents (mg kg^{-1}), % water holding capacity (WHC), organic carbon (OC) and total nitrogen (TN) contents (% by weight), C:N ratios, clay content (% by weight) and pH. CV is the coefficient of variance, SE is the standard error. Significance levels are denoted as follows: * $p < 0.001$, ** $p < 0.005$, *** $p < 0.01$, **** $p < 0.05$.

Figure 1: Cumulative N_2O fluxes (mg N_2O-N m^{-2}) from a) arable and b) grassland soils for 50% water holding capacity (WHC) (50%), ammonium nitrate fertilised 50% WHC (50%+N), 80% WHC (80%) and ammonium nitrate fertilised 80% WHC (80%+N) treatments over a 22 day incubation period. Two-way ANOVA. * $p < 0.001$, ** $p < 0.005$, *** $p < 0.05$.

Figure 2: Emission factors (%) for N_2O emissions from ammonium nitrate fertilised soils (70 kg N ha^{-1}): a) arable soils at 50% and 80% WHC (water holding capacity), b) grassland soils at 50% and 80% WHC. B = Boxworth, G = Gilchriston, R = Rosemaund, W = Woburn, C = Crichton, D = Drayton, Hillsborough = H, N = North Wyke, P = Pwllpeiran.

Figure 3: Cumulative CO_2 fluxes (mg CO_2-C m^{-2}) for a) arable and b) grassland soils for 50% water holding capacity (WHC) (50%), ammonium nitrate fertilised 50% WHC (50%+N), 80% WHC (80%) and ammonium nitrate fertilised 80% WHC (80%+N) treatments over a 22 day incubation period. Measured apparent negative or zero CO_2 fluxes are considered to be due to analytical constraints near the detection limit of the GC.

Figure 4: Cumulative CH_4 fluxes (mg CH_4-C m^{-2}) for a) arable and b) grassland soils for 50% water holding capacity (WHC) (50%), ammonium nitrate fertilised 50% WHC (50%+N), 80% WHC (80%) and ammonium nitrate fertilised 80% WHC (80%+N) treatments over a 22 day incubation period.

Figure 5: a) Arable soil NO_3^--N, b) grassland soil NO_3^--N, c) arable soil NH_4^+-N and d) grassland soil NH_4^+-N contents for untreated pre-incubation (Pre-inc) soils and 50% WHC (50%), ammonium nitrate fertilised 50% WHC (50%+N), 80% WHC (80%) and ammonium nitrate fertilised 80% WHC (80%+N) post-incubation soils. Bars with different letters are significantly different. B = Boxworth, G = Gilchriston, R = Rosemaund, W = Woburn, C = Crichton, D = Drayton, Hillsborough = H, N = North Wyke, P = Pwllpeiran.

Figure 1a)

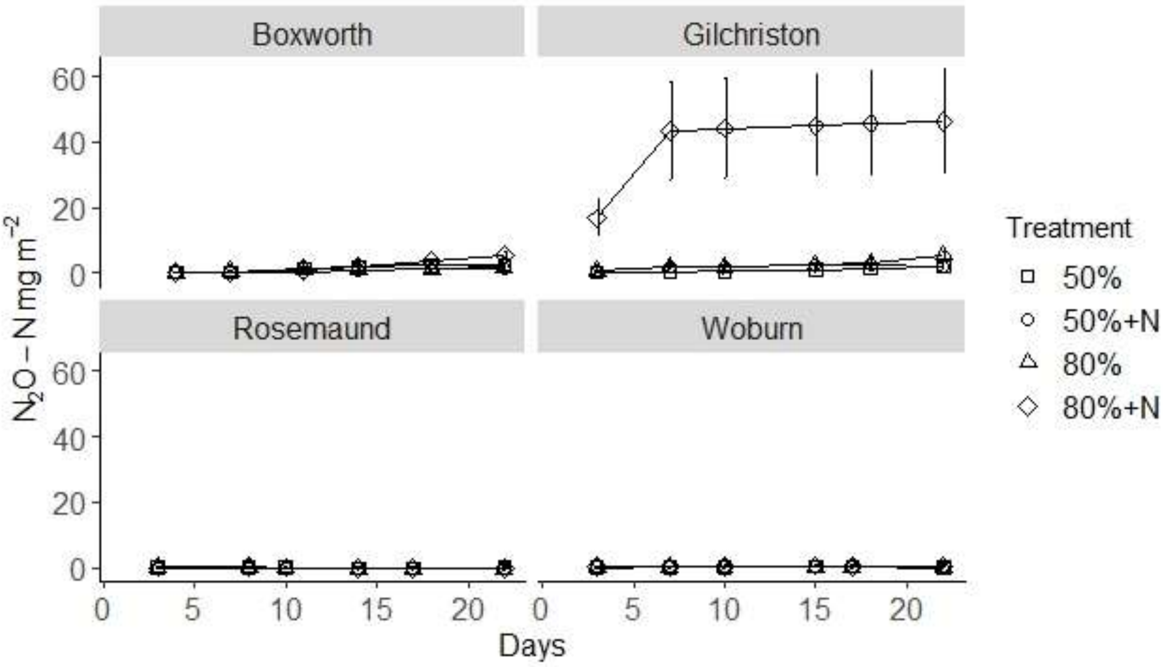


Figure 1b)

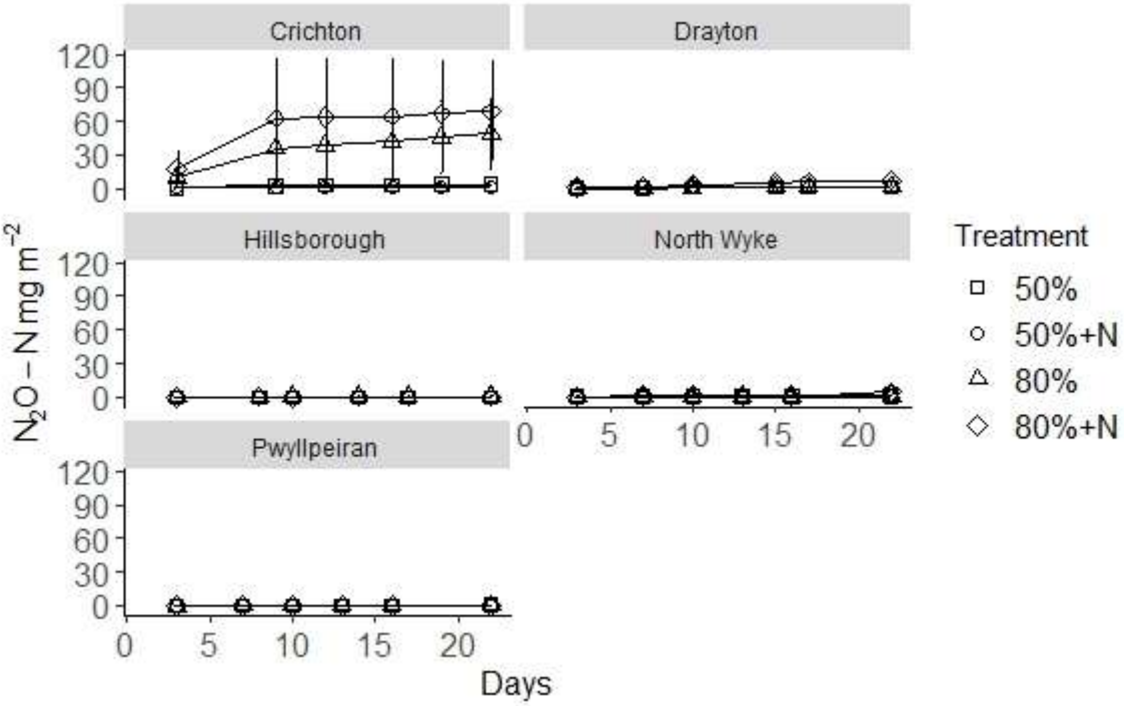


Figure 2)

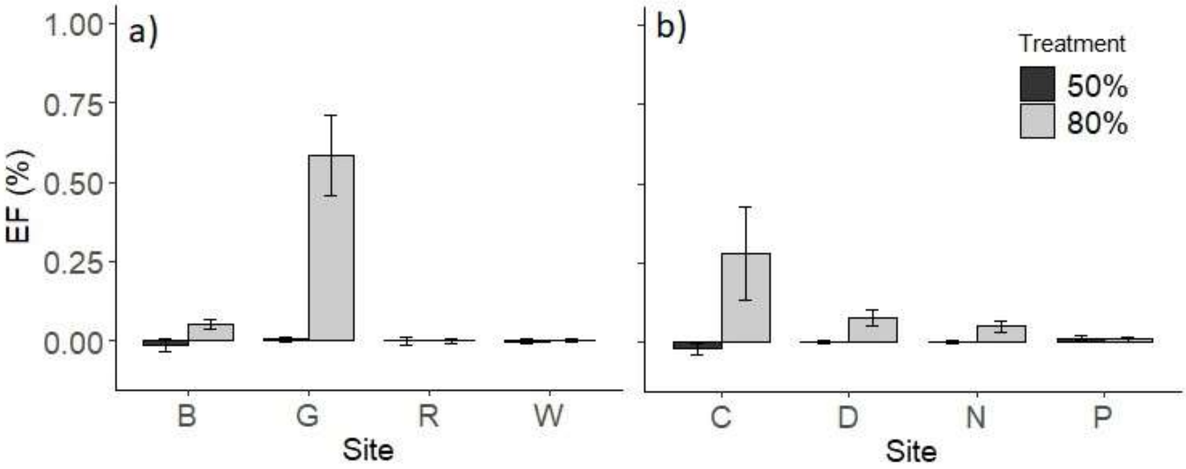
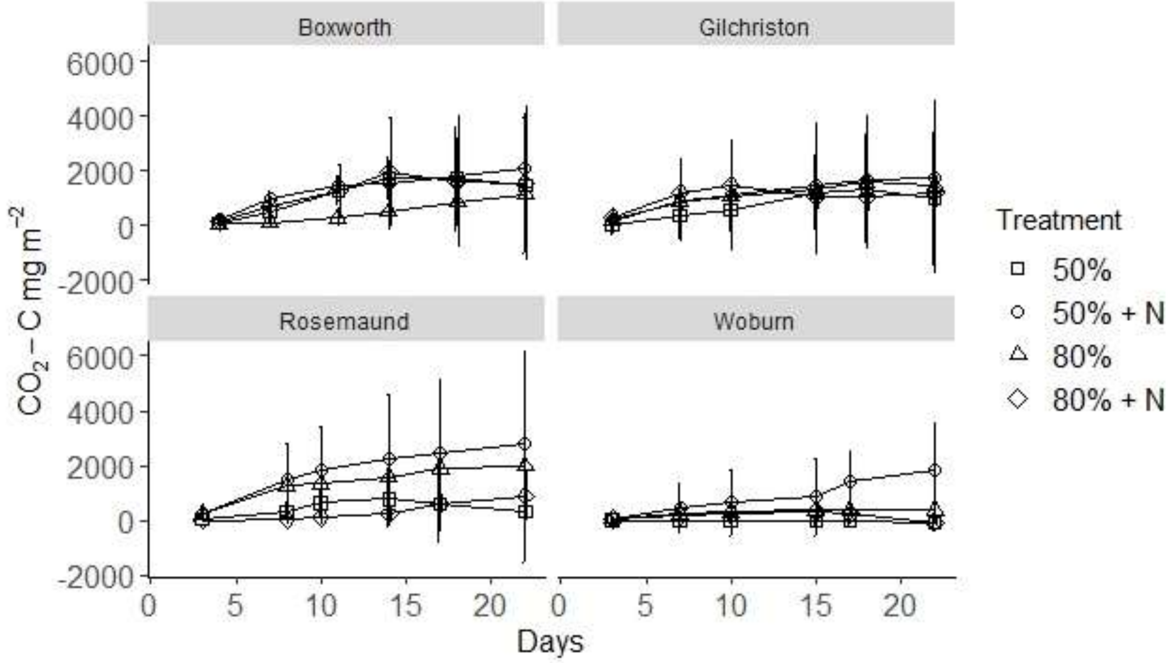


Figure 3a)



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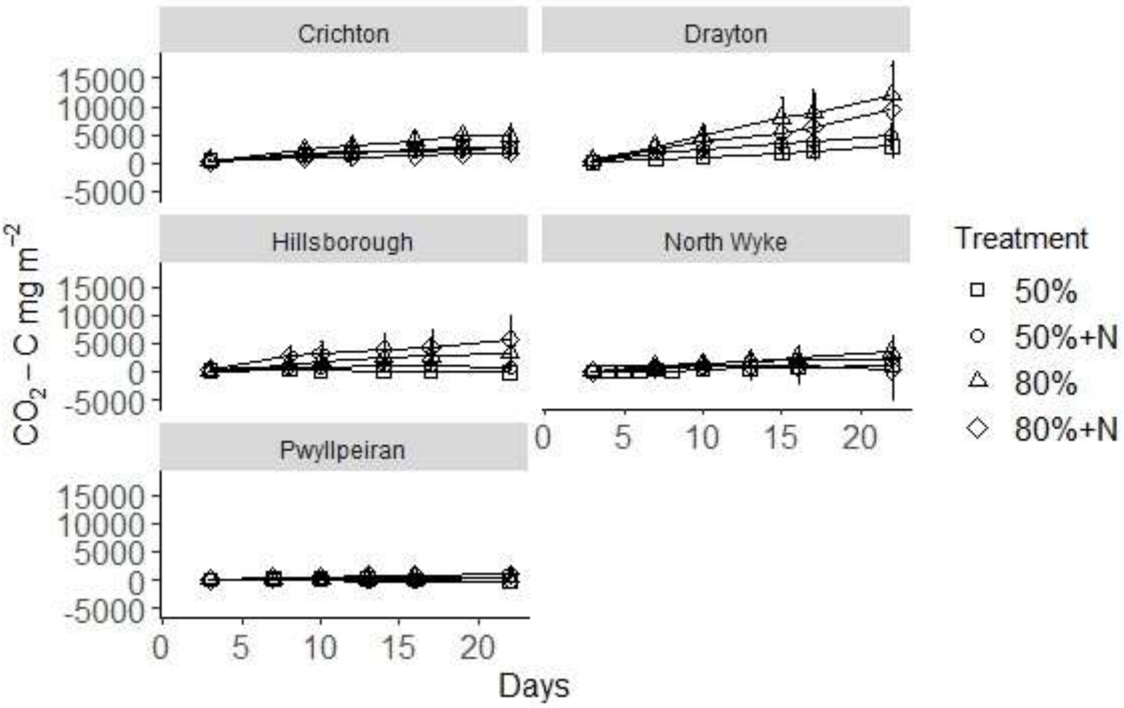

Figure 4a)

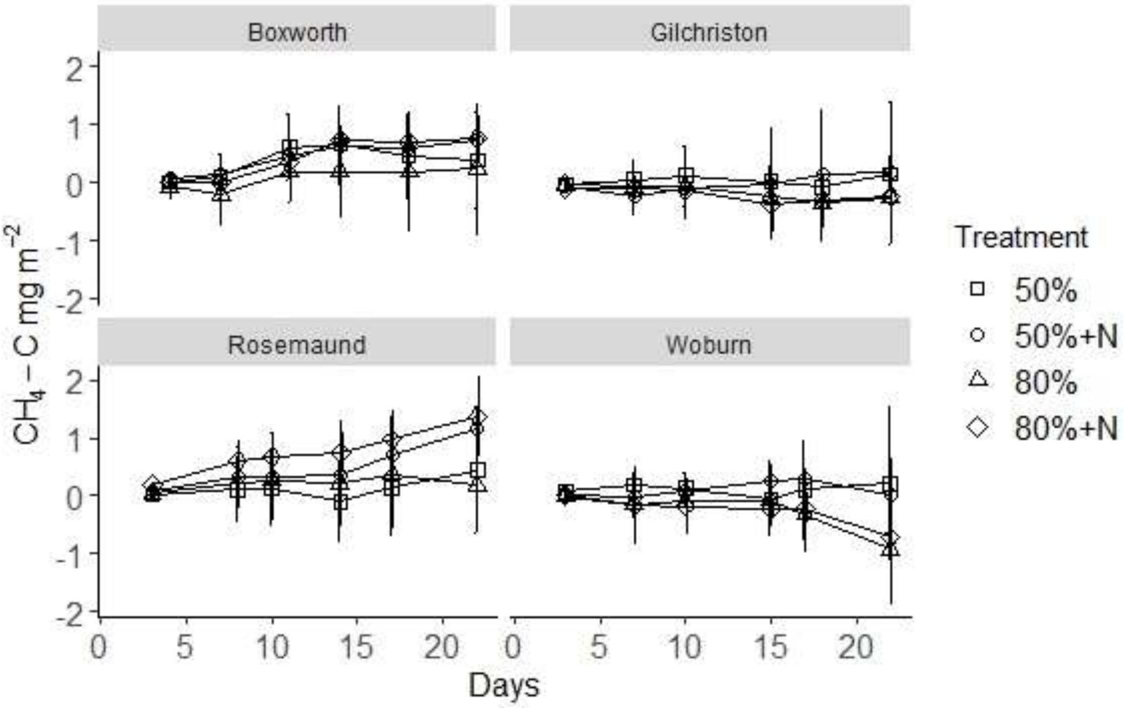

Figure 4b)

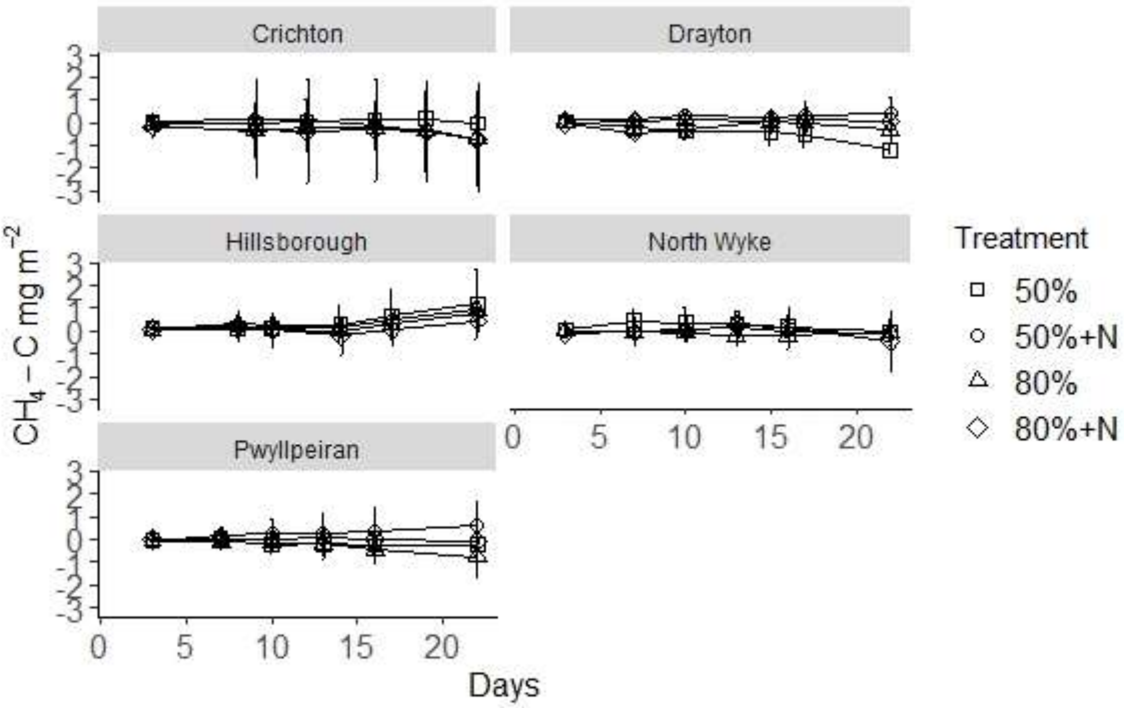

Figure 5)